# Economic valuation of benefits from the proposed REACH restriction of intentionally added microplastics.


Peter King[a],

Alistair Hunt[a], Stavros Georgiou [b], Caroline Rainsford [c] and Richard Dean [d].

[a] University of Bath.

[b] Health and Safety Executive, and University of Reading.

[c] Cosmetic, Toiletry and Perfumery Association.

[d] Environment Agency.



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## 1) Abstract

This study elicited the willingness to pay (WTP) for measures to control the release of intentionally added microplastics. Although microplastics accumulate in the marine environment and are practically unrecoverable, there is considerable scientific uncertainty about their environmental and health effects. This study used both a Choice Experiment (CE) and Contingent Valuation (CV) to evaluate where it was more beneficial to target restrictions at source or emissions. The CE investigated source-control in evaluating how respondents accept a trade-off between the price and performance of cosmetic products when reformulated to reduce the use of microplastics. Two CV tasks then estimated the benefits of research to resolve the uncertainty, and the benefits of upgrading Wastewater Treatment Plants (WWTP) to retain more microplastics. The difference in the annual CV WTP; £53.24 for research and £73.71 respectively, suggests that respondents are willing to pay a substantial premium for the precautionary abatement of microplastics.



**Keywords:** Uncertainty, cost-benefit analysis, stated preference
**JEL Codes:** Q5, Q51, Q58

## 2) Introduction

This paper estimates the non-market benefits of a proposed precautionary restriction on microplastics in the UK. We adopt the definition proposed by the European Chemicals Agency (ECHA) of microplastics as solid, polymer-containing particles smaller than 5mm in size. Microplastics may be intentionally added to consumer products and are substances that may pose environmental and health risks (Lusher, Hollman and Mendoza-Hill, 2017; Lebreton, Egger and Slat, 2019; Thompson et al., 2009). ECHA has proposed a restriction on their use based on their 'extreme' persistence in the environment which proxies for level of risk. The proposed restriction incorporate those sectors that primarily use microplastics such as medicine (single-use and in-vitro products), controlled-release fertilisers (agriculture), detergents, paints, and cosmetics. However, as the cost-effectiveness of the restriction varies with the use, frequency, and type of microplastics used, this research estimates both the benefits of capturing microplastics from all sources at Waste Water Treatment Plants (WWTP), as well as the benefits accruing from the part of the microplastic restriction related to the cosmetic survey. In both cases, the benefits relate to precautionary avoidance of potential environmental and health impacts from intentionally added microplastics, given the lack of certainty over the environmental and health impacts of microplastic.

The potential risks arising from the possibility for microplastics to accumulate in the environment stem from the breakdown of larger plastics in-situ, or their frequent release to the environment either to sewage sludge or to the aquatic environment (Mouat, Lozano and Bateson., 2010; Oosterhuis, Papyrakis and Boteler., 2014; Duis and Coors., 2016; Shen et al., 2020). From sewage, urban run-off or down-the-drain disposal, microplastics may also be found in wastewater (Law and Thompson, 2014). Although WWTP can remove up to 90% of micropollutants from the effluent, compared to the influent, this has limited impact on the environmental concentration of microplastics (Duis and Coors., 2016). Specifically, microplastics remaining in the effluent will filter into the marine environment; an irreversible release. Moreover, the proportion of microplastics that are retained in sewage sludge may still find their way to the terrestrial environment since filtered sewage sludge is often recycled in the UK (ECHA, 2019). Once in the environment, Lebreton, Egger and Slat (2019), building on Lebreton et al. (2018) and Jambeck et al. (2005), discuss the distribution of marine plastics, both macro and micro-sized, and report that a significant proportion would persist around coastal areas, for example, through stranding, settling, sinking below the surface, or transportation into the open ocean where they may accumulate, where the challenge of deep-sea exploration limits scientific knowledge about their effects (Hammer, Kraak and Parsons, 2012; Derraik, 2002; Abate et al., 2020). Microplastics that accumulate in the marine environment may then be ingested by marine life and, as indicated by Kosuth, Mason and Wattenberg (2018), possibly by humans via seafood or water. Although a range of physical and chemical health effects have been hypothesised or observed in some marine life, there is currently no evidence to suggest that levels of human ingestion of marine microplastics have deleterious consequences (Lusher, Hollman and Mendoza-Hill, 2017). However, with the growth in plastic production and accumulation of microplastics, the absence of any health effect on humans may not persist (Lebreton et al., 2018). The current low level of impacts and the possibility of higher environmental and health risks from marine microplastics may not preclude restrictions on the use of microplastics under the precautionary principle. This principle states that the lack of full scientific certainty, such as the scientific uncertainty about the existence and magnitude of possible environmental and health impacts of microplastic ingestion, shall not be a barrier to regulation (Kuntz-Duriseti, 2004; Gollier and Treich, 2003).

This study provides monetary valuations of the benefits of a future possible restriction on intentionally added microplastics in the UK. A novel feature of this paper is that we estimate the nonmarket benefits to both public and private approaches to reducing the release of microplastics. Furthermore, we contribute to the debate on whether abatement should target source-control or effluent filtering. In the following sections, we first provide a brief overview of the general approach to economic valuation in the context of the precautionary principle. We then review the small but emerging literature on the valuation of micropollutants. We then detail the valuation methods used in the present study with a focus on the survey design, responses, and results. The final section reports the elicited valuations and discusses the implications for both the environmental, economic literature and regulators.

# 3) Literature Review

The benefits of restrictions to improve the quality of the marine environment can be valued at what individuals are Willing-To-Pay (WTP) for potential improvement. However, there are several challenges in the literature regarding the validity of aggregating WTP for use in Cost-Benefit Analysis (CBA) in the presence of uncertainty and irreversibility (Kuntz-Duriseti, 2004). Specifically, Ha-Duong (1998) suggested that regulators could choose either immediate restrictions to avoid the irreversible loss of pollutants or delay action to resolve the uncertainty. Uncertainty and irreversibility are both evident properties of the microplastic problem as there is a significant degree of scientific uncertainty about the environmental and health impacts of marine microplastics. Furthermore, the release of microplastics is irreversible, given that they are practically unrecoverable once lost to the environment. This paper builds upon previous studies that have sought to elicit WTP to avoid health and environment risks from chemical contaminants (e.g. ECHA, 2016; Adamowicz et al., 2011).

However, as the micropollutants and marine pollution fields are relatively new, there is a paucity of SP literature specifically valuing micropollutants. The most relevant valuations in the literature come from Abate et al. (2020) and Logar et al. (2014) as they used similar methods in related scenarios, while Choi and Lee (2018), and Kim et al. (2019) also discussed WTP to reduce marine pollution. This section reviews previous use of SP methods to value microplastic reductions. Logar et al. (2014) undertook a choice experiment (CE) to assess individual WTP for improvements in WWTP to filter micropollutants. The resulting WTP was $73 per household per year to reduce the potential environmental risk of micropollutants to a negligible level via changes to WWTP. Micropollutants, though more toxic, share many of the same sources and transport mechanisms as microplastics. Logar et al. (2014) also acknowledged the role of scientific uncertainty in the field of micropollutants, although they chose not to focus on microplastics specifically.

A study carried out for the Environment Agency (EA, 2015) used a CE to elicit WTP to avoid potential human health risks from the pollutants Octamethylcyclotetrasiloxane, (D4), and Decamethylcyclopentasiloxane (D5) (EA, 2015). Whilst D4/5 are unrelated to microplastics, there are parallels to this paper. Specifically, D4/5 have uses in personal care products and household care products such as cleaning detergents, similar to microplastics. Furthermore, ECHA (2019) noted that.

*"Environmental (and other) benefits arise from the reduction in potential risks associated with the accumulation of D4/D5 in the aquatic environment."*

Unlike microplastics, EA (2015) noted that while D4 satisfies the definitions for a Persistent Bioaccumulative and Toxic (PBT) substance and D5 satisfies the definition of a Very Persistent and Very Bioaccumulative (vPvB) substance, both are characterised by their irreversible accumulation in water (EA, 2015). Microplastics are themselves inert, and many studies have investigated their potential to act as a vector for toxic contaminants, with variable results

(Koelmans et al., 2016; Burns and Boxall., 2018). The EA (2015) decision to use a CE to elicit WTP reflects a choice to model the substitution between the three attributes; product quality and how it changes with and without D4/5, product prices, and the reduction in accumulation and potential risks from contaminants. The choice of three attributes is consistent with the modern CE literature summarised in Lancsar and Louviere (2008), Hoyos (2010), Ryan, Gerard, and Amaya-Amaya (2007).

CV methods have also been used to estimate the value of reductions in microplastics. For instance, Abate et al. (2020) evaluated a proposal in Norway to reduce marine plastics in Arctic ice. Their representative Norwegian sample of 552 (response rate: 25%) reported that the average annual household WTP to reduce all Arctic marine plastic was $642 (approximately £517). Their CV question was as follows.

*"Considering the anticipated results of the initiative outlined before, would you vote for this initiative if the initiative would cost your household an annual tax of NOK XXX for the next ten years?"*

Abate et al. (2020) initially planned to use a CE, but pre-testing revealed that it was challenging to accurately design a scenario given the scientific uncertainty. When designed, respondents found it too complex and did not consider all the attributes. Therefore, they opted for the CV format to elicit the total value of benefits.

A final paper that uses CV is Choi and Lee (2018) who reported an average annual WTP of $2.59 per person with total annual household WTP for the Seoul population of $9.8 million. Their results suggest that respondents were willing to pay to reduce water-borne endocrine-disrupting substances. Their scenario was product labelling financed through an annual tax, in contrast to our scenario of a restriction on microplastics financed through product prices or water bills.

## a) Methodology

Table 1: CE Example:

| | Option A | Option B |
|---|---|---|

| | | |
|---|---|---|
| Reduction in the effectiveness of the personal care product. | 0% | 5% |
| Percentage reduction in the release of microplastics from cosmetics. | 0% | 10% |
| Increase in product price. | £0 | £1 |
| I prefer: | | |

Choice Experiments (CE) ask respondents to select their preferred alternative, which is described by a series of attributes that vary by the levels they take (Hoyos, 2010). Attributes, such as price and quantity, are used based on Lancaster's (1966) characteristic theory of value which posits that goods are considered as a combination of their characteristics or attributes. The attributes take different levels depending on the scenario; examples being the status-quo or the proposed level of abatement. Respondents assumed to be utility-maximising, then select an alternative with levels of attributes that maximises their utility. Where a price attribute is included, respondents' marginal rate of substitution between attributes may be interpreted as attribute specific WTP for marginal changes in attribute levels (OECD, 2006; Bateman et al., 2002).

An example of the specific CE design used in this research is reproduced in Table 1. The two alternatives correspond to an opt-out status quo (Option A) without changes, and a scenario similar to the ECHA restriction (Option B). The status quo option is required for consistency with theory while we choose only one alternative scenario for simplicity (Bateman et al., 2002). The two alternatives differ only in the levels of the three attributes; two (percentage reduction in performance and percentage reduction in release) with three levels and one attribute (product price) with four levels. The number of attributes was chosen following pre-testing, and each was described in depth to respondents before answering. The effectiveness attribute later referred to as "performance ", had three levels; (5, 10 and 50%) and was expected a priori to have a negative sign. The reduction in release attribute referred to as "emission ", also took three levels, (10, 40, 90%) and was expected to have a positive sign as respondents were expected to value improved environmental quality. This attribute is similar to the "potential environmental risk" attributes used in Logar et al. (2014). Finally, the price attribute took four levels (£1, 5, 10, 20) with an expected negative sign. The CE used an orthogonal main-effects d-efficient design. There were 16 different choice sets, and each respondent completed one randomly assigned block of four choices. Blocking of the sets is used to minimise task complexity. Before answering any of the four tasks, respondents were given a dominated scenario test where one alternative was unambiguously superior to the other. Respondents failing this test of attention and understanding were excluded from the sample. Overall, a CE is used to evaluate how consumers would tradeoff attributes of personal care products in response to an ECHA restriction on the use of microplastics in cosmetics.

i) CVM format

CV is a commonly used SP method where respondents state their individual WTP valuations for nonmarket valuation problems. The design of the specific payment question is an area of substantial debate. The most common CVM question format is Dichotomous Choice (DC) which is akin to the referenda format that the seminal N.O.A.A panel recommended. DC methods ask respondent to vote yes or no to a proposed scenario with entailed cost. Choi and Lee (2018) used the single-bound DC (SBDC) format to report an average annual WTP of $2.59 per person for product labelling on products that contain microplastics. The SBDC format is generally preferred in the literature given its superior incentive-compatibility. However, the double-bound DC (DBDC) format, which asks a follow-up question after the SBDC, reports more information about respondent's preferences at the expense of possible anchoring effects. In practice, the first stage of the DBDC question appears to respondents as a SBDC format, which is incentive-compatible, as it is presented individually with no indication of follow-up bids. When respondents are presented the second stage, they then have no option to update their previous valuation. Therefore, the DBDC format may have similar incentive-compatibility while eliciting more precise WTP. This research adopts first SBDC and then DBDC CVM formats to elicit WTP for two different scenarios.

The first CVM question used in this research used the SBDC format to elicit WTP to undertake research into the potential environmental and health effects of microplastics. This scenario represented the value of resolving uncertainty whilst irreversible releases of microplastics continue. The text of this question is given below and stresses the trade-off of research improving future knowledge but having no immediate effect.

**Q6** *"One possible policy option would be to fund research into the long-term environmental and health effects of microplastics in the environment.*

*The research would definitely resolve the scientific uncertainty about any possible effects, though it would have no effect on the amount of microplastics currently entering the environment from wastewater sewerage.*

*An increase in your water bills would cover only the cost of this research. Any follow up action, depending on the research findings, would be funded separately. Would your household be willing to pay £X per year in extra water bills specifically for such research?"*

The question is therefore a single-bounded question with bid levels randomly varying: {5, 10, 20, 40, 60, 80, 90, 100}. 364/670 respondents answered this question first. The payment vehicle of water bills was used after pre-testing to improve how realistic the scenario was.

The second CVM question (here Q7) elicited WTP for a reduction in the release of intentionally added microplastics given upgrades to Wastewater Treatment Plants (WWTP); although no research would be undertaken. We argue that the difference in WTP between the two CVM scenarios indicates whether respondents' value restrictions that are precautionary in nature. The sample was split in two with approximately half answering this question first and the other half answering the second question. As half of the sample were asked this SBDC question first, respondents could not anchor their valuations on previous tasks. The ordering of the two questions is randomly varied in the survey to mitigate ordering effects.

**Q7** [1] *"Suppose that the UK was going to introduce a policy that would stop microplastics from wastewater sewerage entering the environment now, before waiting for the results of the research discussed in the previous question.*

*This policy would pay to upgrade wastewater treatment plants filtering systems so that they would capture all the microplastics in sewerage wastewater heading to the environment.*

*An increase in your water bills would be used to pay for the cost of this investment. Would your household be willing to pay £X per year in extra water bills to implement this policy?"*

The question used a double-bound question with bid levels randomly varying from:

Q7: {5, 10, 20, 40, 60, 80, 90, 100}

Q7B (upper bound): {10, 20, 40, 80, 120, 160, 180, 200}

Q7C (lower bound): {2.50, 5, 10, 20, 30, 40, 45, 50}.

b) Survey design

The survey instrument was designed in conjunction following the Johnston et al. (2017) guidance for SP surveys and a rigorous pre-testing process that included expert consultation and a pilot study. The survey instrument consisted of five sections comprising socioeconomic questions, the CV questions, the CE, questions assessing environmental attitudes and finally a small section of debriefing questions. The survey design is somewhat novel in the literature in using both CV and CE sections to evaluate a public and private good approach to microplastic restrictions.

c) Data Collection

In April 2020, DJS Research LTD collected 670 responses to the survey, reflecting a response rate of 65%. Table 2 shows that this compares favourably to comparable literature both in size and response rate. There were 25 separate questions which took an average respondent 7.5 minutes to complete. DJS were asked to randomly vary both the CV bid levels and the CE block that respondents were assigned to. The sample of 670 was truncated to 304 using six different rules. The first three dealt with survey participation. Firstly, `speeders' who completed the survey faster than 50% of the median, were excluded (Scasny and Zvěřinová., 2014). Survey understanding below 7/10, and protestors (identified and categorised using text responses) were also excluded (Rakotonarivo, Schaafsma and Hockley, 2016). The final three truncation rules had theoretical support. Firstly, those failing the dominated scenario were excluded for irrational preferences or misunderstanding the task (Foster and Mourato, 2002). As a necessary condition for incentive-compatible responses, respondents had to

[1] The wording here is taken from the Q6 then Q7 ordering, and so mentions previous valuation scenarios. The text is suitably amended in the reversed order. Furthermore, the text is amended in the second question to justify asking a second valuation question. Finally, note that both scenarios are UK-specific and so cannot easily be aggregated to the larger and more diverse EU measure.

report the survey to be policy consequential (did not assess payment consequentiality) (Vossler and Watson., 2013). Finally, those reporting uncertainty about their choices were removed. Including all six rules reduced the sample size to 304/670. The characteristics of the sample are reported in Table 3 and suggest that the sample was broadly representative of the UK adult population. Sample income is marginally lower than the population, possibly owing to gender differences or having more students in the sample.

Table 2: Sample sizes and response rates for four similar SP studies of water quality or micropollutants.

| Study | Sample Size | Response Rate |
|---|---|---|
| This study (April 2020) | 670 | 68% |
| Abate et al. (2020) | 552 | 25% |
| Jørgensen et al. (2013) | 754 | 37% |
| Logar et al. (2014) | 1000 | 25% |
| Adamowicz et al. (2011) | 1292 | 46% |

Table 3: Full sample characteristics

| Category | Sample | Population |
|---|---|---|
| **Gender** | Male: 46% Female: 53% | Male: 49% Female: 51% |
| **Age** | Mean: 42 years old | Mean: 38 years old |
| **Trips** | Annual trips: 1.6 | Annual coastal trips: ~4 |
| **Charity donations** | Donated: 33%<br>Not: 63% | Data from the Charities Aid Foundation (2019) suggests that more than 60% of people are involved with charities.<br>However, environmental charities make up fewer than 4% of total charitable donations. |
| **Education** | A level or equivalent: 50.75%<br>Graduate or more: 49.25% | A level or equivalent: 40.4%<br>Graduate or more: 42% |
| **Employment** | Prefer not to say = 2.68%<br>NEET = 11.34%<br>Retired = 7.76%<br>Student = 4.47%<br>Part-time = 14.95%<br>Self-employed = 6.85%<br>Full-time = 51% | NEET: 3.7%<br>Student: 3.5%<br>Part-time: 10.4%<br>Self-employed: 15.1%<br>Full-time: 36.9% |
| **Income** | Mean: £2,138 | Mean: £2,340.<br>Due to more students and a younger sample. |

Sources: The data sources are the Natural England (2016) Report that monitored engagement with the Natural Environment and as such is an appropriate source to validate the figures. Data for employment and income data come from the Office for National Statistics.

## 2) Choice Experiment Results

The CE results were estimated[2] using first Multinomial Logit (MNL) and then Mixed Logit (MXL) models. Equation 1 is the indirect utility function for respondent *I* who gains utility from each attribute. The subscript *o* is for option A or B. Note that the socioeconomic variables reported in Table 5 are interacted with the alternative-specific constant. All three attributes are assumed to be the negative lognormally distributed random parameters using 1000 Halton draws. The model is estimated in WTP-space which implies that the MRS between attributes is directly interpretable as WTP (Train and Weeks, 2005).

$$V_{io} = ASC_o + \beta_{Price} * (Price_o + \beta_{Performance} * Performance_o + \beta_{Emission} * Emission_o) \qquad 1$$

$$\beta_{Price_i} = -exp\left(\mu_{log(\beta_{Price})} + \sigma_{log(\beta_{Price})} * \xi_{Price,i}\right) \qquad 2$$

The results of different specifications are displayed in Table 4. The table reports MWTP and goodness-of-fit statistics to justify the model choice. The best model, given minimised log-likelihood and AIC, is the MXL in WTP-space on truncated sample with all covariates and with the three attributes all randomly distributed negative lognormally; this is reported in Table 5. The columns reporting marginal WTP for a one-percentage reduction in the release of microplastics (here called 'Emission') ranged from £0.028 to £0.038. The MWTP for the one-percentage reduction in effectiveness (called 'Performance') ranged from (-£0.036 to -£0.060). The final accepted CE MWTP is £0.035 for a one percent reduction in microplastics lost from a personal care product and a Marginal Willingness to Accept (MWTA) of £0.036 for a one percent loss of performance.

Table 4: CE MWTP and goodness-of-fit by model specification.

| Specifications | Emission MWTP | Performance MWTP | Sample Size | AIC | R2 | LogLikelihood |
|---|---|---|---|---|---|---|
| Conditional Logit | 0.038 | -0.043 | 670 | 3539.16 | 0.035 | -1765.60 |
| MNL: Full sample | 0.035 | -0.044 | 670 | 3428.99 | NA | -1694.493 |
| MNL: Truncated | 0.032 | -0.060 | 304 | 1462.88 | NA | -712.4113 |
| MXL: Attributes only | 0.028 | -0.046 | 670 | 2974.91 | 0.203 | -1480.455 |
| MXL: Full sample | 0.037 | -0.043 | 670 | 2961.54 | 0.214 | -1458.768 |
| MXL: Truncated | 0.035 | -0.036 | 304 | 1290.48 | 0.183 | -623.2379 |

[2] Estimated using the MLOGIT and APOLLO packages for R. Replication code available on request.

The model reported in Table 5 reports statistically significant coefficients for each attribute of the CE. This indicates that respondents were sensitive to changes in their levels. There were two justifications for distributing each attribute negative log-normally. Firstly, the lognormal distribution has been used repeatedly in the applied CE literature (Stackelberg and Hammitt, 2009; Ghosh, Maitra and Das,2013; Train and Weeks, 2005; Zambrano-Monserrate and Ruano, 2020)  and is appropriate where the normal distribution is not behaviourally plausible; especially for the price parameter. A second justification is that while MWTP and goodness-of-fit were best when distributing all parameters negative lognormally. The mean parameters of the three attributes represent MWTP for a one percent change in the attributes. The MWTP is robust to using the full sample as is the significance and signs of the covariates. The MWTP for each attribute, for both samples and with confidence intervals is reported in Table 5. Respondents are willing to pay a similar amount for marginal reductions in the release of microplastics from personal care products. This suggests that respondents would tradeoff product attributes and be willing to pay a premium for microplastic-free products if a restriction were enforced. However, given that MWTP was estimated in per-product terms it is not possible to assert whether such source-controls would then be economically viable.

The standard deviations of the attributes were highly statistically significant which indicates preference heterogeneity around the mean coefficient of the attribute at the respondent-level. The statistically significant standard deviations motivate the use of mixed logit instead of conditional or multinomial logit which impose the assumptions of preference homogeneity. The model also reports an Alternative Specific Coefficient (ASC) defined against Option B following the utility-difference approach. However, it was statistically insignificant, which indicates no strong bias towards either option (Train, 2009). A further area to discuss the MXL is the covariates. With regard to question order, the dummy has an insignificant effect which may indicate that providing respondents with the precautionary option of Q7 immediately before the CE did not influence their choices. The insignificant variable on the choice task respondents were allocated to supports the blocking of the 16 possible choice tasks into four blocks. Beliefs in the policy consequentiality of the scenario also had statistically significant positive effects on choices which motivates its use as a truncation rule. As expected, income was weakly statistically significant. Finally, belief in experts also exerted a statistically significant effect on respondent choices. Belief in experts may be linked to respondents' perceptions of the expert-recommended information provided before the CE. The insignificance of many socioeconomic characteristics including age, gender, trips, education is unusual in the literature. To summarise, this MXL, reports improved goodness-of-fit, and produces plausible MWTP and, therefore, is the preferred specification for modelling respondent heterogeneity from the CE.

Table 5: MXL model in WTP-space, truncated sample and all covariates.

| **Variable** | **Estimate** | **Bootstrap.std.err.** | **Bootstrap.t.ratio(0)** | **Bootstrap.p-val(0)** |
|---|---|---|---|---|
| ASC | -0.370 | 2.465 | -0.150 | 0.427 |
| $\mu_{\{Price\}}$ | -0.228*** | 0.237 | -0.964 | 0.000 |
| $\sigma_{\{Price\}}$ | 1.544*** | 0.444 | 3.480 | 0.000 |
| $\mu_{\{Performance\}}$ | -0.036*** | 0.235 | -15.534 | 0.000 |
| $\sigma_{\{Performance\}}$ | -1.406*** | 0.092 | -15.329 | 0.000 |
| $\mu_{\{Emission\}}$ | 0.035*** | 0.087 | -41.133 | 0.000 |
| $\sigma_{\{Emission\}}$ | 2.519*** | 0.095 | 26.388 | 0.000 |
| $\beta_{\{Q1Gender\}}$ | -0.219 | 0.413 | -0.531 | 0.265 |
| $\beta_{\{Q2Age\}}$ | 0.014 | 0.021 | 0.690 | 0.185 |
| $\beta_{Q3Distance}$ | -0.006 | 0.013 | -0.453 | 0.300 |
| $\beta_{Q4Trips}$ | -0.082 | 0.312 | -0.262 | 0.358 |
| $\beta_{Q12CECertainty}$ | -0.360 | 0.408 | -0.882 | 0.160 |
| $\beta_{Q16BluePlanet}$ | -0.246 | 0.361 | -0.681 | 0.188 |
| $\beta_{Q18Charity}$ | -0.125 | 0.517 | -0.242 | 0.366 |
| $\beta_{Q20Consequentiality}$ | 0.798** | 0.422 | 1.889 | 0.017 |
| $\beta_{Q21Experts}$ | 0.516** | 0.302 | 1.706 | 0.014 |
| $\beta_{Q22Education}$ | 0.142 | 0.319 | 0.445 | 0.247 |
| $\beta_{Q23Employment}$ | 0.174** | 0.123 | 1.414 | 0.045 |
| $\beta_{Q24Income}$ | -0.000* | 0.000 | -0.981 | 0.096 |
| $\beta_{Q25Understanding}$ | -0.229 | 0.222 | -1.032 | 0.107 |
| $\beta_{Order}$ | 0.084 | 0.447 | 0.189 | 0.405 |
| $\beta_{Task}$ | 0.035 | 0.103 | 0.345 | 0.353 |
| **Estimation Statistics** | | | | |
| Adjusted R2: | | | 0.1834 | |
| Likelihood ratio test: | | | chisq = 579.29 (p = 0.000) | |
| AIC | | | 1290.48 | |
| Log-likelihood | | | -623.2379 | |
| **Attribute** | **Lower bound** | **Truncated sample** | **Upper bound** | **Full sample** |
| Emission | £0.034 | £0.036 | £0.039 | £0.038 |
| Performance | -£0.043 | -£0.037 | -£0.057 | -£0.045 |

## 3) Contingent Valuation

A range of models is used to elicit WTP from the two CV tasks with specification, WTP and goodness-of-fit criteria reported in Table 6. The median unit values are £53.25 for question six (research, no reduction in pollution) and £73.71 for question seven (reduction in pollution, no research). Median values are preferred over mean to reduce the effect of outliers and where values may be lognormally distributed (Zambrano-Monserrate and Ruano, 2020). Table 6 reports that WTP is sensitive to question format and econometric model. For instance, a consistent ordering effect is evident such that WTP values are higher if Q6 is before Q7. Furthermore, the substantial difference between the WTP where respondents believed the survey to be (or not to be) consequential for policy supports the use of consequentiality as a rule for truncating the sample. The fact that "inconsequential" values are found to be lower than the "consequential" values is consistent with empirical results suggesting that those who believe their participation to be inconsequential would report deflated valuations (Vossler and Watson, 2013., Czajkowski et al., 2017). It may be that respondents who did not believe their survey participation to be consequential voted against any payment and thus the values are lower. Generally, models including covariates fit the data better while bid-only models are appropriate for estimating sample median WTP. Finally, the effect of including the follow-up questions in the analysis is to reduce sample WTP.

The econometric model for the CV data is given below. Firstly, respondent's utility $u$ is a function of the deterministic component with income $y$ and socioeconomic characteristics $X$, a stochastic component $\varepsilon$ which represents unobserved factors. The probability of a respondent $i$ answering `Yes' that they are able and willing to pay a stated bid level b for the policy proposal depends on the comparison between the utility of the policy and cost $(U_{i1})$ and status quo utility $(U_{i0})$ in Equation 3. In Equation 4, $\Phi$ is the standard normal with $\sigma$ the standard deviation of the error term $\varepsilon$ which leaves $\beta, \gamma$ to be estimated (Abate et al., 2020; Manga, Oru and Ngwabie, 2019; Alberini and Scasny, 2014). We estimate a bid-only model to elicit WTP and then a full-covariates model to better model the determinants of WTP.

CV Choice Probability: $P(Yes_i, b_i) = P[U_{i1}(y_i - b_i, X_i, \varepsilon_{i1}) U_{i0}(y_i, X_i, \varepsilon_{i0}) >]$ 3

Probit Model: $P(Yes_i | X_i, b_i) = \Phi\left(\frac{\beta X_i}{\sigma} - \frac{\gamma}{\sigma} b_i\right)$ 4

Bid-only model to elicit sample WTP: $Q6Response_i = \alpha_i + Q6Bid_i + \varepsilon_i$ 5

Q6 SBDC Model with all covariates: $Q6Response_i = Q6Bid_i + X_i + \varepsilon_i$ 6

Probit WTP: $meanWTP = -\frac{\alpha_i}{\gamma}$ 7

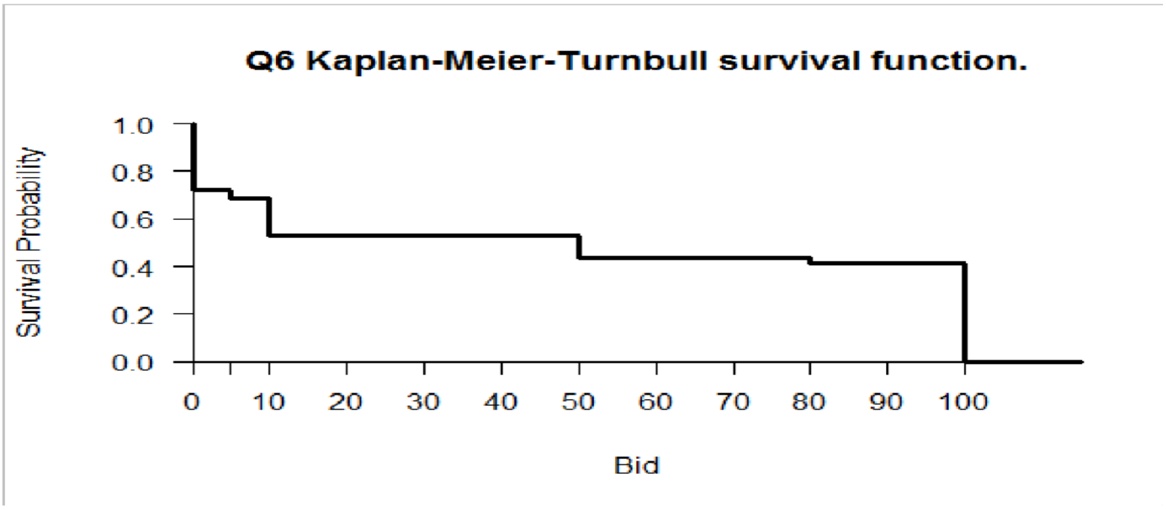
Table 6: CVM WTP from each sample, order and measure. Adopted models in italics.

| **Specifications** | **WTP** | **N** | **AIC** | *R*2 | **Log-likelihood** |
|---|---|---|---|---|---|
| **Q6** | | | | | |
| Bid-Only (Full-sample) | £53.25 | 670 | 7103 | 0.04 | -3549.52 |
| *Bid-Only (Truncated-sample)* | *£53.25* | *304* | *2968* | *0.06* | *-1482.45* |
| Covariates (Full sample) | £53.23 | 670 | 6156.99 | 0.18 | -3061.49 |
| Covariates (Truncated sample) | £50.54 | 304 | 2325 | 0.272 | -1150 |
| Consequential | £61.83 | 560 | 5128.65 | 0.176 | -2548.32 |
| Inconsequential | £21.32 | 110 | 911.572 | 0.243 | -439.78 |
| Order1 | £51.76 | 364 | 2651.87 | 0.227 | -1309.94 |
| Order2 | £51.58 | 306 | 3289.31 | 0.192 | -1628.65 |
| **Q7: single-bound only** | | | | | |
| Bid-Only (Full-sample) | £88.43 | 670 | 6637.86 | 0.05 | -3316.90 |
| Bid-Only (Truncated-sample) | £86.13 | 304 | 2623 | 0.09 | -1309.87 |
| Covariates (Full sample) | £91.39 | 670 | 6003.08 | 0.14 | -2984.54 |
| Covariates (Truncated sample) | £89.79 | 304 | 2361 | 0.171 | -1163.77 |
| Consequential | £96.02 | 560 | 4808 | 0.150 | -2392.14 |
| Inconsequential | £45.99 | 110 | 1131.42 | 0.0922 | -553.71 |
| Order1 | £94.00 | 364 | 2681.47 | 0.151 | -1327.74 |
| Order2 | £87.09 | 306 | 3189.48 | 0.176 | -1581.74 |
| **Q7: double-bound** | | | | | |
| Bid-Only (Full-sample) | £85.21 | 670 | 14922.55 | 0.001 | -7549.28 |
| *Bid-Only (Truncated-sample)* | *£73.71* | *304* | *5723* | *0.001* | *-2859* |
| Covariates (Full sample) | £73.71 | 670 | 14009.67 | 0.173 | -6987.83 |
| Covariates (Truncated sample) | £83.46 | 304 | 5329 | 0.190 | -2649 |

a) Question 6 CVM

The distribution of the WTP for the first CVM task can be analysed using the Kaplan-Meier-Turnbull (KMT) survival function. Carson and Steinberg (1990) developed KMT as a nonparametric extension to the Kaplan-Meier survival function. KMT is appropriate for data whereby responses lie within bid intervals (Lim, Jin and Yoo, 2017). Figure 1 shows that a bid of zero pounds was accepted with 100% probability while the highest bid level of £100 has an approximately 41% probability of being accepted. The interpretation here is that the probability of a respondent accepting a proposed bid level falls with the increase in the bid value as expected in theory.

Figure 1: The survival function estimator for the probability of accepting the Q6 bid levels.




The first CVM scenario used the SBDC format and can, therefore, be analysed using Probit. Table 8 reports the univariate probit with covariates for the question six model. Most covariates had a statistically significant impact on WTP. As expected, the bid level is highly significant as are order and timing which supports the evidence of an ordering effect and supports the exclusion of speeders. The small effect size of the income coefficient may be due to the survey asking for gross monthly income which is a far larger value than the binary 0,1 decision in the models. However, income was still statistically significant as expected. There are two other notable results from the covariates. Firstly, a distance-decay effect cannot entirely be ruled out in this research. This is because while distance from the coast was not statistically significant, the number of trips to the coast was highly significant. Secondly, both charity involvement (defined as donations or membership), and viewership of the Blue-Planet II programme were highly statistically significant determinants of respondent’s behaviour. Their positive and relatively large marginal effects indicate that involvement in pro-environmental media and activities influences WTP. The question six model indicates that respondents were willing to pay £53.25 per-household per-year in extra water bills for research to resolve the scientific uncertainty about microplastics. The median value is used to reduce the effect of outliers.

Table 8: This table reports bootstrapped Q6 CVM WTP using probit (N = 304).

| **Variable** | **Estimate** | **Marginal Effect** | **Std. Error** | **Pr(>\|z\|)** |
|---|---|---|---|---|
| (Intercept) | -2.487 | | 0.207 | 0.000*** |
| Order | -0.053 | -0.017 | 0.062 | 0.400 |
| Q1Gender | -0.108 | -0.024 | 0.063 | 0.089* |
| Q2Age | -0.001 | 0.000 | 0.003 | 0.561 |
| Q3Distance | 0.0001 | 0.000 | 0.002 | 0.802 |
| Q4Trips | 0.066 | 0.015 | 0.033 | 0.041** |
| Q16BP | 0.458 | 0.112 | 0.049 | 0.000*** |
| Q18Charity | 0.560 | 0.135 | 0.057 | 0.000*** |
| Q21Experts | 0.604 | 0.144 | 0.037 | 0.000*** |
| Q22Education | -0.036 | -0.010 | 0.031 | 0.247 |
| Q23Employment | 0.193 | 0.047 | 0.018 | 0.000*** |
| Q24AIncome | 0.0001 | 0.000 | 0.000 | 0.042** |
| Timing | -0.000 | 0.000 | 0.000 | 0.001*** |
| BID | -0.018 | -0.004 | 0.001 | 0.000*** |

| **Goodness-of-fit:** | |
|---|---|
| Log-likelihood: | -3075.887 |
| Pseudo $R^2$: | 0.1718 |
| LR test p value: | 0.000 |
| AIC: | 6179.77 |

**WTP:**

| **Measure** | **Lower** | **Mean** | **Upper** |
|---|---|---|---|
| Mean | £66.74 | £70.50 | £75.04 |
| truncated Mean | £50.05 | £51.40 | £52.83 |
| adjusted truncated Mean | £69.42 | £73.11 | £76.99 |
| Median | £50.11 | £53.25 | £56.55 |

b) Question 7 CVM

Q7 asked respondents what they would be willing to pay to resolve the irreversible release of microplastics by upgrading WWTPs for improved microplastic capture; a scenario similar to Logar et al. (2014). Figure 2 reports two KMT survival functions; one for the SBDC part of the question, the second with the follow-up included in the DBDC format. The follow-up bids are used to gain more information about the distribution of WTP.

Figure 2: The survival function estimator for the probability of accepting the Q7 bids: SBDC top graph and DBDC bottom graph.

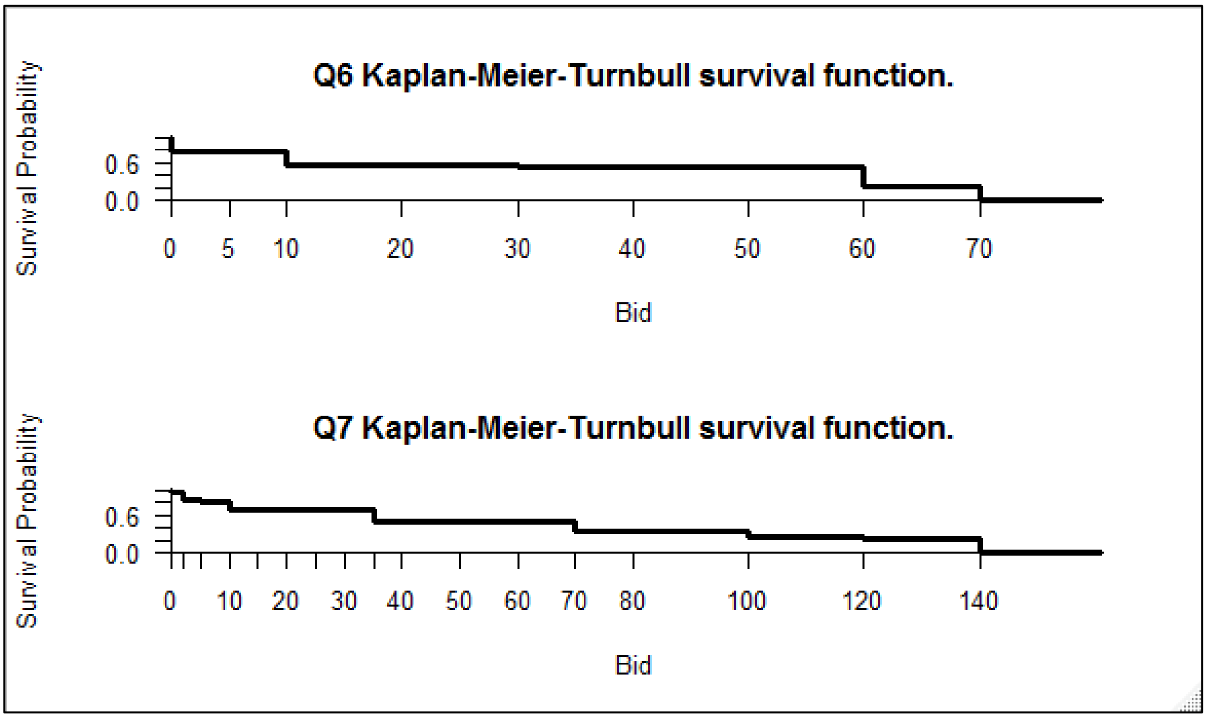


Table 10: Bootstrapped Q7 CVM WTP using bivariate probit (N = 304).

| **Variable** | **Estimate** | **Marginal Effect** | **Std. Error** | **Pr(>/z/)** |
|---|---|---|---|---|
| (Intercept) | -1.466 | | 0.174 | 0.000*** |
| Order | 0.104 | 0.047 | 0.055 | 0.057* |
| Q1Gender | -0.106 | -0.029 | 0.055 | 0.055* |
| Q2Age | -0.007 | -0.003 | 0.002 | 0.001** |
| Q3Distance | 0.005 | 0.002 | 0.002 | 0.001*** |
| Q4Trips | 0.060 | 0.022 | 0.028 | 0.030** |
| Q16BP | 0.192 | 0.068 | 0.043 | 0.000*** |
| Q18Charity | 0.303 | 0.012 | 0.050 | 0.000*** |
| Q21Experts | 0.248 | 0.098 | 0.031 | 0.000*** |
| Q22Education | 0.021 | 0.016 | 0.027 | 0.428 |
| Q23Employment | 0.093 | 0.032 | 0.016 | 0.000*** |
| Q24AIncome | 0.000 | -0.000 | 0.000 | 0.000*** |
| Timing | 0.001 | 0.001 | 0.000 | 0.000*** |
| BID | -0.021 | -0.001 | 0.000 | 0.000*** |

| **Goodness-of-fit:** | |
|---|---|
| Log-likelihood: | -7193.28 |
| Pseudo $R^2$: | 0.1733 |
| LR test p value: | 0.000 |
| AIC: | 14414.51 |

| **WTP** | | | |
|---|---|---|---|
| **Measure** | **Lower** | **Mean** | **Upper** |
| Mean | £80.43 | £83.01 | £85.63 |
| truncated Mean | £77.53 | £79.71 | £81.99 |
| adjusted truncated Mean | £82.49 | £85.41 | £88.42 |
| Median | £70.94 | £73.71 | £76.55 |

Table 10 presents the analysis of the WTP values derived from Question 7 in the survey. Similar to the previous CVM models, the coefficients on ordering, timing and bid levels were highly significant, as expected. This model reports slightly improved $R^2$ but much poorer log-likelihood and AIC, possibly due to the DBDC format using more information. Furthermore, income was statistically significant as expected but reported a very weak marginal effect; possibly due to including the gross monthly average income rather than a dummy on whether a respondent was above or below median income. Moreover, environmental indicators such as viewership of the Blue-Planet or involvement in environmental charities were also highly statistically significant determinants of respondents CVM preferences. Finally, when comparing the two CV tasks, respondents report higher WTP for precautionary abatement via enhanced filtering, a trend robust to ordering and consequentiality beliefs.

c) Precautionary WTP.

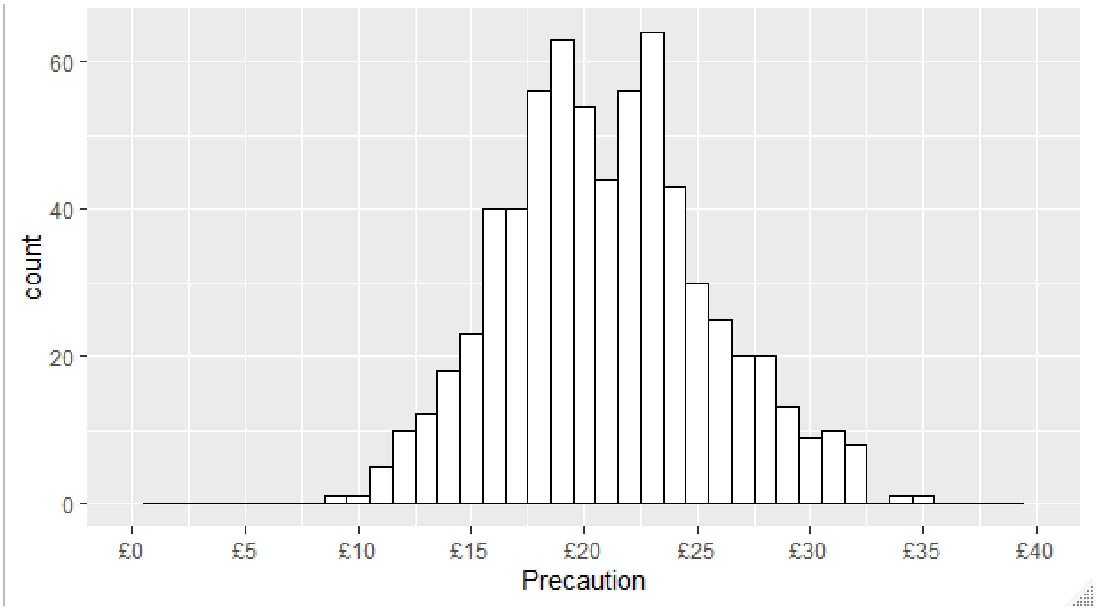
The difference in respondent WTP between the two (Q7 minus Q6 WTP) may be interpreted as a premium for any precautionary action to reduce the release of microplastics. The concept of a precautionary premium was first discussed in Kuntz-Duriseti (2004) as an economic interpretation of the precautionary principle. However, their interpretation focused on a premium to compensate respondents for the effect of uncertainty rather than a difference in WTP for scenarios. The premium in our research is shown in Figure 3. The histogram reports that there were few respondents with relatively small (<£10) or relatively large premia (>£30). Although there are two peaks of the distribution, an average respondent reported a difference of £20.99 between the two questions which may be interpreted as the additional increase in benefits to precautionary restrictions. Therefore, the use of two CV scenarios results in support for immediate abatement policies. More generally, this research suggests that respondents prefer to resolve irreversibility over resolving uncertainty.

## 4) Discussion.

### a) Cost-Benefit Analysis.

The WTP elicited in this research can be used to inform policy on the targeting of restrictions either at source (such as requiring the reformulation of microplastics), or at emission (by investing in WWTP filtration). The CV WTP was estimated per-household given that the payment vehicle, water bills, are paid at the household level. The CVM WTP can then be

Figure 3: Histogram of respondent precaution premium.




aggregated to estimate the total benefit of upgrading WWTP by using the number of UK households in 2019 (27.8 million) and the median WTP of £73.71 (£70.94 - £76.55). This method reports total annual WTP in the UK of £2.05 billion (£1.97 bn - £2.12bn). How this compares to the cost of upgrading WWTP is unclear for three reasons. Firstly, the cost of upgrading WWTP is highly plant-specific with some plants already able to retain a high proportion of microplastics in wastewater yet others without any tertiary filtration systems. Secondly, a variety of options exist to upgrade filtration systems and choosing one specifically for microplastics may not be optimised for another pollutant. Finally, retaining more microplastics from wastewater means more microplastics in sewage sludge. Sludge is often recycled to the terrestrial environment, unless subject to thermal destruction, in which case microplastics may still be distributed to the environment. Howevere, indicative costs of upgrading WWTP can be estimated by scaling the estimates from Eunomia (2019).. They estimated the cost of upgrading WWTP in Europe to tertiary filtration was €1.49 billion (at time of writing: £1.37billion). As the spatial scope is larger than the proposal for the UK, the values have to be scaled to the UK. However, it is unclear how many WWTP in the UK would require upgrades and what technology would be used to upgrade them. Therefore, it is possible that it is economically viable to upgrade WWTP to retain more microplastics thus targeting restrictions at the emissions stage. However, the viability may be highly sensitive to changes in upgrade costs.

Estimating the benefits of source-control are more challenging given that the WTP elicited in this research is for a marginal 1% change in product attributes, and it is unclear whether WTP can be assumed to be proportional to larger percentage changes. Furthermore, the MWTP is also elicited on a per-product basis, and aggregation would require knowledge of the average price of individual personal-care products, the average amount of personal care products purchased per person annually in the UK and an understanding of the relationship between product-performance and microplastic use in individual products. This relationship is not only product-specific, as some products such as sun cream are required to maintain their stated levels of protection, but also firm-specific as different firms will be differentially able to invest in research and development to reformulate their products without microplastics. Our results therefore serve primarily to highlight the possibility of deriving benefit estimates using stated preference methods; subsequent research is needed to refine these estimates to the specific decision context. Overall, it is therefore unclear whether targeting restrictions at source or emission is more economically viable.

## b) Effect of coronavirus on WTP.

To understand the effect of undertaking the data collection during the coronavirus (COVID-19) pandemic, a question was added to the survey which asked respondents whether their income was affected by the pandemic (answers: Yes, No, prefer not to say). The data collection period could not be changed as there was no certainty as to when the effects of the pandemic would abate. Moreover, the temporal stability of preferences is unclear (Bateman and Langford, 1997). Although the included question was an imperfect measurement, initial results suggest that the pandemic affects WTP through income effects. Indeed, nearly half (47.6%) of the sample reported that their income had been affected by the pandemic. Furthermore, the pandemic, job-losses and economic worries were all cited as reasons for protest votes in the sample. Table 11 reports fitted CE (at the individual level using the delta-method from the MXL) and CV WTP (individual level from Probit using

Krinsky-Robb bootstrap) by whether respondents reported that their income was affected by the pandemic (`Affected' column) and whether their income was above or below the median (`Income group' column)[3]. There is a noticeable income effect whereby `high' income leads to higher WTP in the CV WTP regardless of whether they were affected. Moreover, being affected led to lower-income and thus, lower WTP. Curiously, the CE WTP changed signs at the highest effect and income levels, although it is unclear why this is

Table 11: COVID-19 effects on WTP across both CE and CV tasks.

| Affected | Income group | N | Performance | Emission | Q6 | Q7 |
|---|---|---|---|---|---|---|
| No | Low | 175 | 0.155 | 0.133 | 22.99 | 44.09 |
| No | High | 160 | 0.070 | 0.059 | 24.19 | 46.20 |
| Yes | Low | 160 | 0.060 | 0.052 | 23.97 | 43.67 |
| Yes | High | 179 | 0.041 | 0.035 | 23.95 | 44.99 |

## c) Limitations

The limitations of this study relate to the design and scale of the SP survey. For example, our limited sample size precluded our randomising the order for the CVM-CE sections. A related limitation is the relatively small split samples, exacerbated by the high truncation rate. The truncation rules have strong theoretical justification, so the high exclusion rate suggests that the design complexity was high despite the high median level of survey understanding and the pre-testing process. Truncation is shown to have a small negative effect on both CE and CV WTP and the survey design's validity. Additionally, there was a statistically significant difference in WTP across the split-samples which corroborates the finding that multiple CV scenarios are not independent draws from the individual distribution of WTP (Kjær et al., 2006; Day et al.,2012). WTP from each subsample was recovered and shows that ordering has a small negative effect on the magnitude of WTP. A plausible explanation could be that the more questions respondents answer, the less willing they were to accept a given CV scenario (Day et al., 2012).

## d) Summary

From a representative sample of 670 UK adults, this study elicited WTP for three different measures to control the loss of intentionally added microplastics to the terrestrial and marine environments. The CE section of the SP survey asks respondents about their preferences for the price, performance, and reduction of microplastics from their personal care products. The CE scenario was developed with input from the CTPA and EA and thus has credibility with respect to its information content. As such, the sample WTP is £0.036 for a one percentage point decrease in the number of microplastics lost per product, and WTA of £0.048 for a one

[3] The subsamples are similarly sized and do not sum to 670 as some respondents had exactly median income or preferred not to answer.

percentage point loss of product performance. Although the two values cannot easily be aggregated and scaled, the comparison of the two shows that respondents value product performance highly, as expected, although the WTP for reductions in the environmental impacts of personal care products is also substantial. Finally, the two CVM questions indicate a strong individual preference, expressed as a premium in the WTP valuations, for precautionary restrictions on the irreversible release of microplastics to the environment instead of delaying regulation to resolve the significant uncertainty around microplastics. Future work in this area can address the scientific uncertainty such as the extent of any environmental or human health risks, as well as the relationship between microplastic use and performance of personal-care products, and the value of restricting microplastics in other sectors which emit more microplastics. Future work may use these two CVM tasks to empirically estimate Quasi-Option Value (QOV). QOV is the value to delaying a decision to facilitate learning and, therefore, can be appended to CBA to incorporate the issues of irreversibility and uncertainty (Traeger, 2014). Mensink and Requate (2005) argue that QOV is simply the Option Value (OV) for receiving information, thus QOV can be interpreted as the expected value of future information. Estimating the value of information can inform CBA as to whether to regulate immediately or delay to gather more information. To estimate QOV, future work may use two CVM scenarios where both have reductions in micropollutants but only one scenario also resolves the uncertainty.

# Bibliography

Abate, T.G., Börger, T., Aanesen, M., Falk-Andersson, J., Wyles, K.J. and Beaumont, N., 2020. Valuation of marine plastic pollution in the European arctic: Applying an integrated choice and latent variable model to contingent valuation. *Ecological Economics*, 169, p.106521. Available from: https://www.sciencedirect.com/science/article/pii/S0921800919307360?casa_token=_HIJenc-K_EAAAAA:StNMBVy6YtT2FCsLHxTtx_r-VpFShjYOOGINVAogOtj7muhkfUCV7g8hiIeXnqIdsSm28d1V4g

Adamowicz, W., Dupont, D., Krupnick, A. and Zhang, J., 2011. Valuation of cancer and microbial disease risk reductions in municipal drinking water: An analysis of risk context using multiple valuation methods. *Journal of Environmental Economics and Management*, 61(2), pp.213–226. Available from: https://www.sciencedirect.com/science/article/pii/S0095069610001063?casa_token=wXcH_CTM-l8AAAAA:ghZgTf9TBo3KQzJrEjVGifcUNwKmjTt5qBBDRIs1RyDm6SBpzJkViXVpZDAeV6RfCLYnO1_oWg

Alberini, A., Boyle, K. and Welsh, M., 2003. Analysis of contingent valuation data with multiple bids and response options allowing respondents to express uncertainty. *Journal of environmental economics and management*, 45(1), pp.40–62. Available from: https://www.sciencedirect.com/science/article/pii/S0095069602000104?casa_token=bfC6iL52h68AAAAA:dGzhAUM41AP7g8ig6XlKDPK0v4qJJTWa9NJ6oaKlDOc7lwIKSqioKpAK01wXNQBCtnxqnTRykw

Alberini, A. and Scasny, M., 2014. *Stated-preference study to examine the economic value of benefits of avoiding selected adverse human health outcomes due to exposure to chemicals in the European union: Part iii: Carcinogens*. European Chemicals Agency. Available from: https://echa.europa.eu/documents/10162/13630/study_economic_benefits_avoiding_adverse_health_outcomes_3_en.pdf/ccf1db6e-fa50-45e3-a0b4-178a05cc8b96.

Arrow, K., Solow, R., Portney, P., Leamer, E., Radner, R. and Schuman, H., 1993. Report of the N.O.A.A panel on contingent valuation. *Federal Register*, 58(10), pp.4601–4614. Available from: https://www.researchgate.net/profile/Edward_Leamer/publication/277297107_Kenneth_Arrow/links/572a241108ae2efbfdbc1959/Kenneth-Arrow.pdf.

Azzarello, M. and Van-Vleet, E., 1987. Marine birds and plastic pollution. *Marine Ecology Progress Series*, 37, pp.295–303. Available from: http://www.int-res.com/articles/meps/37/m037p295.pdf.

Barboza, Luís Gabriel Antão, et al. "Marine microplastic debris: An emerging issue for food security, food safety and human health." Marine pollution bulletin 133 (2018): 336-348. Available from: https://www.sciencedirect.com/science/article/pii/S0025326X1830376X

Bateman, I.J., Carson, R.T., Day, B., Hanemann, M., Hanley, N., Hett, T., Jones-Lee, M., Loomes, G., Mourato, S., Pearce, D.W. et al., 2002. *Economic valuation with stated preference techniques: A manual*. Edward Elgar Publishing Ltd. Available from: https://www.cabdirect.org/cabdirect/abstract/20043091507.

Bateman, I.J., Cooper, P., Georgiou, S., Navrud, S., Poe, G.L., Ready, R.C., Riera, P., Ryan, M. and Vossler, C.A., 2005. Economic valuation of policies for managing acidity in remote mountain lakes: Examining validity through scope sensitivity testing. *Aquatic Sciences*, 67(3), pp.274–291. Available from: https://link.springer.com/content/pdf/10.1007/s00027-004-0744-3.pdf.

Bateman, I.J. and Langford, I.H., 1997. Budget-constraint, temporal, and question ordering effects in contingent valuation studies. *Environment and Planning A*, 29(7), pp.1215–1228. Available from: https://journals.sagepub.com/doi/abs/10.1068/ a291215.

Beaumont, N.J., Aanesen, M., Austen, M.C., Borger, T., Clark, J.R., Cole, M., Hooper, T., Lindeque, P.K., Pascoe, C. and Wyles, K.J., 2019. Global ecological, social and economic impacts of marine plastic. *Marine pollution bulletin*, 142, pp.189–195. Available from: https://www.sciencedirect.com/science/article/ pii/S0025326X19302061.

Blomquist, G.C., Blumenschein, K. and Johannesson, M., 2009. Eliciting willingness to pay without bias using follow-up certainty statements: comparisons between probably/definitely and a 10-point certainty scale. *Environmental and Resource Economics*, 43(4), pp.473–502. Available from: http://gattonweb.uky.edu/faculty/blomquist/ERE%202009%20Aug%20Blomquist%20Blumenschein%20Johannesson.pdf.

Boardman, A.E., Greenberg, D.H., Vining, A.R. and Weimer, D.L., 2017. *Cost-benefit analysis: concepts and practice*. 4th ed. Prentice Hall. Available from: https://www.nrc.gov/docs/ML1233/ML12338A400.pdf.

Brouwer, R., Hadzhiyska, D., Ioakeimidis, C. and Ouderdorp, H., 2017. The social costs of marine litter along European coasts. *Ocean & coastal management*, 138, pp.38–49. Available from: https://www.sciencedirect.com/science/article/pii/S0964569117300297#bib43.

Burns, E.E. and Boxall, A.B., 2018. Microplastics in the aquatic environment: Evidence for or against adverse impacts and major knowledge gaps. Environmental toxicology and chemistry, 37(11), pp.2776-2796. Available from: https://setac.onlinelibrary.wiley.com/doi/full/10.1002/etc.4268

Cameron, T.A., 2005. Individual option prices for climate change mitigation. *Journal of Public Economics*, 89(2-3), pp.283–301. Available from: https://www.sciencedirect.com/science/article/pii/S0047272704000131.

Carson, R.T. and Steinberg, D., 1990. Experimental design for discrete choice voter preference surveys. 1989 Proceeding of the Survey Methodology Section of the American Statistical Association, pp.821– 822. Available from: https://www.researchgate.net/profile/Dan_Steinberg2/publication/246138041_Experimental_Design_for_Discrete_Choice_Voter_Preference_Surveys/links/567dcf8408ae051f9ae493ff/Experimental-Design-for-Discrete-Choice-Voter-Preference-Surveys.pdf.

Charities Aid Foundation, 2019. CAF UK giving 2019. an overview of charitable giving in the UK. London: Charities Aid Foundation. Available from: https://www.cafonline.org/docs/default-source/about-us-publications/ caf-uk-giving-2019-report-an-overview-of-charitable-giving-in-the-uk. pdf?sfvrsn=c4a29a40_4

Chen, Q., Allgeier, A., Yin, D. and Hollert, H., 2019. Leaching of endocrine disrupting chemicals from marine microplastics and mesoplastics under common life stress conditions. Environment international, 130, p.104938. Available from: https://pubmed.ncbi.nlm.nih.gov/31234003/

Choi, E.C. and Lee, J.S., 2018. The willingness to pay for removing the microplastics in the ocean–the case of Seoul metropolitan area, south Korea. *Marine Policy*, 93, pp.93–100. Available from: https://www.sciencedirect.com/science/article/ pii/S0308597X17304803.

Cosmetics Europe, 2020. Cosmetics Europe's Response To The Consultation On The Draft SEAC Opinion To Support Annex XV Restriction Report.

Cummings, R.G., Harrison, G.W. and Rutström, E.E., 1995. Homegrown values and hypothetical surveys: is the dichotomous choice approach incentive-compatible? *The American Economic Review*, 85(1), pp.260–266. Available from: https://www.jstor.org/stable/pdf/2118008.pdf.

Day, B., Bateman, I.J., Carson, R.T., Dupont, D., Louviere, J.J., Morimoto, S., Scarpa, R. and Wang, P., 2012. Ordering effects and choice set awareness in repeat-response stated preference studies. *Journal of environmental economics and management*, 63(1), pp.73–91. Available from: https://www.sciencedirect.com/science/article/pii/ S0095069611001239.

DEFRA, 2017. *Implementation of the environmental protection (microbeads) (England) regulations 2017*. London: DEFRA. Available from: https://www.legislation.gov.uk/ukia/2017/160/pdfs/ukia_20170160_en.pdf.

Derraik, J., 2002. The pollution of the marine environment by plastic debris: a review. *Marine pollution bulletin*, 44(9), pp.842–852. Available from: https:

//www.sciencedirect.com/science/article/pii/S0025326X02002205.

DFA, 2006. *Handbook of cost-benefit analysis*. Department of Finance and Administration, Commonwealth of Australia. Available from: https://www.finance.gov.au/sites/default/files/Handbook_of_CB_analysis.pdf.

Duis, K. and Coors, A., 2016. Microplastics in the aquatic and terrestrial environment: sources (with a specific focus on personal care products), fate and effects. *Environmental Sciences Europe*, 28(1), p.2. Available from: https://enveurope.springeropen.com/articles/10.1186/s12302-015-0069-y.

ECHA, 2014. *Stated-preference study to examine the economic value of benefits of avoiding selected adverse human health outcomes due to exposure to chemicals in the European union: Part i: sensitization & dose toxicity*. European Chemicals Agency. Available from: https://echa.europa.eu/documents/10162/13630/study_economic_benefits_avoiding_adverse_health_outcomes_1_en. pdf/5c884e27-8ce7-4ffd-97dc-5e7875ee67e0.

ECHA, 2016. *Valuing selected health impacts of chemicals*. European Chemicals Agency. Available from: https://echa.europa.eu/documents/10162/13630/echa_review_ wtp_en.pdf.

ECHA, 2019. *Annex xv restriction report proposal for a restriction*. European Chemicals Agency. Available from: https://echa.europa.eu/documents/10162/ 12414bc7-6bb2-17e7-c9ec-652a20fa43fc.

Environment Agency (2015), "A Choice Experiment Valuation of Environmental Precaution:

Cyclotetrasiloxane (D4) and Cyclopentasiloxane (D5)", Unpublished Report.

Eunomia, 2019. Investigating options for reducing releases in the aquatic environment of microplastics emitted by (but not intentionally added in)products. Available from: https://www.eunomia.co.uk/reports-tools/investigating-options-for-reducing-releases-in-the-aquatic-environment-of-microplastics-emitted-by-products/

Foster, V. and Mourato, S., 2002. Testing for consistency in contingent ranking experiments. *Journal of environmental Economics* and *Management*, 44(2), pp.309–328. Available from: https://reader.elsevier.com/reader/sd/pii/S0095069601912033?token=352EAE3A2CFF7E970277F3C2E770E5AE004F5B26E02E253856C48F5A8121E2831845D37286257DF4D552C

Fransson, N. and Garling, T., 1999. Environmental concern: Conceptual definitions, measurement methods, and research findings. *Journal of environmental psychology*, 19(4), pp.369–382. Available from: https://www.researchgate.net/profile/Niklas_Fransson/publication/222252867_Environmental_Concern_Conceptual_Definitions_Measurement_

Methods_and_Research_Findings/links/5a0c5866a6fdcc39e9bf6438/
Environmental-Concern-Conceptual-Definitions-Measurement-Methods-and-Research-Findin
pdf.

Gollier, C. and Treich, N., 2003. Decision-making under scientific uncertainty: the economics of the precautionary principle. *Journal of Risk and Uncertainty*, 27(1), pp.77–103. Available from: https://link.springer.com/content/pdf/10.1023/A: 1025576823096.pdf.

Ha-Duong, M., 1998. Quasi-option value and climate policy choices. *Energy Economics*, 20(5-6), pp.599–620. Available from: https://www.sciencedirect.com/science/ article/pii/S0140988398000115.

Hammer, J., Kraak, M. and Parsons, J., 2012. Plastics in the marine environment: the dark side of a modern gift. *Reviews of environmental contamination and toxicology*, pp.1–44. Available from: https://www.researchgate.net/profile/ John_Parsons3/publication/225049699_Plastics_in_the_Marine_Environment_ The_Dark_Side_of_a_Modern_Gift/links/0912f5072f10dd513a000000/ Plastics-in-the-Marine-Environment-The-Dark-Side-of-a-Modern-Gift.pdf.

Hanemann, M., Loomis, J. and Kanninen, B., 1991. Statistical efficiency of double bounded dichotomous choice contingent valuation. *American journal of agricultural economics*, 73(4), pp.1255–1263. Available from: https://www.jstor.org/stable/pdf/1242453.pdf?casa_token=meY1RIjw77EAAAAA:xzLDWSLx9b5qur72f5E8 _bDrizqFrp0-KPrFmnYcrcx2Ds8XAmd9waqWfUckL-Kc- QZ5MmIKe3hiNKV6jg8cgHvSp9_a4zR6JFoQnj6apgOiYooO-hQ

Hanemann, W.M.,1984. Discrete/continuous models of consumer demand. *Econometrica: Journal of the Econometric Society*, pp.541–561. Available from: https://www.jstor.org/stable/pdf/1913464.pdf?casa_token= vlPRHYKGmjgAAAAA:I_jNVIvphbSABPAxCbOTxQLsTLJ19gjdakZRaRbQgj76fHOKUbI_ dP3QQvlMvcL9K9tvwdk1uUxA5UgG6ReFB2yz7eCIXqgemJmsSVuph0GzTJJpgKY.

Hausman, J.A. and McFadden, D., 1981. Specification tests for the multinomial logit model. *Cambridge, Mass.: Dept. of Economics, Massachusetts Institute of Technology*. Available from: https://www.jstor.org/stable/pdf/1910997.pdf?refreqid= excelsior%3A6b7093d9993b0f1b1a6cb49099f07acf.

Hooper, T.L., 2013. *Evaluating the costs and benefits of tidal range energy generation*. Ph.D. thesis. University of Bath. Available from: https://purehost.bath.ac.uk/ ws/portalfiles/portal/187948106/Hooper_Tara_PhD_Thesis_17_12_13.pdf

Hoyos, D., 2010. The state of the art of environmental valuation with discrete choice experiments. *Ecological economics*, 69(8), pp.1595–1603. Available from: https://www.sciencedirect.com/science/article/pii/S0921800910001540#bib73.

HSE, 2015. *Annex xv restriction report proposal for a restriction: Octamethylcyclotetrasiloxane*. Health and Safety Executive. Available from: https://echa.europa.eu/ documents/10162/9a53a4d9-a641-4b7b-ad58-8fec6cf26229.

Jambeck, J., Geyer, R., Wilcox, C., Siegler, T., Perryman, M., Andrady, A., Narayan, R. and Law, K., 2005. Plastic waste inputs from land into the ocean. *Science*, 347(6223), pp.768–771. Available from: http://science.sciencemag.org/content/sci/347/ 6223/768.full.pdf.

Jâms, I.B., Windsor, F.M., Poudevigne-Durance, T., Ormerod, S.J. and Durance, I., 2020. Estimating the size distribution of plastics ingested by animals. *Nature communications*, *11*(1), pp.1-7. Available from: https://www.nature.com/articles/s41467-020-15406-6

Johnston, R., Boyle, K., Adamowicz, W., Bennett, J., Brouwer, R., Cameron, T., Hanemann, W., Hanley, N., Ryan, M., Scarpa, R. and Tourangeau, R., 2017. Contemporary guidance for stated preference studies. *Journal of the Association of Environmental and Resource Economists*, 4(2), pp.319–405. Available from: http://aura.abdn.ac.uk/bitstream/handle/2164/10529/691697.pdf?sequence=1.

Jørgensen, S.L., Olsen, S.B., Ladenburg, J., Martinsen, L., Svenningsen, S.R. and Hasler, B., 2013. Spatially induced disparities in users' and non-users' WTP for water quality improvements—testing the effect of multiple substitutes and distance decay. *Ecological Economics*, 92, pp.58–66. Available from: https://www.sciencedirect.com/science/article/pii/S0921800912002790?via%3Dihub#s0030.

Kim, H.J., Lee, H.J. and Yoo, S.H., 2019. Public willingness to pay for endocrine disrupting chemicals-free labelling policy in Korea. *Applied Economics*, 51(2), pp.131–140. Available from: https://www.tandfonline.com/doi/full/10.1080/00036846.2018.1494803.

Kjær, T., Bech, M., Gyrd-Hansen, D. and Hart-Hansen, K., 2006. Ordering effect and price sensitivity in discrete choice experiments: need we worry? *Health economics*, 15(11), pp.1217–1228. Available from: https://onlinelibrary.wiley.com/doi/pdf/10.1002/hec.1117.

Koelmans, A.A., Bakir, A., Burton, G.A., Janssen, Colin, R., 2016a. Microplastic as a Vector for Chemicals in the Aquatic Environment Critical Review and Model-Supported Reinterpretation of Empirical Studies. Environ. Sci. Technol. 50, 3315–3326. Available from: https://pubs.acs.org/doi/pdf/10.1021/acs.est.5b06069

Kollmuss, A. and Agyeman, J., 2002. Mind the gap: why do people act environmentally and what are the barriers to pro-environmental behaviour? *Environmental education research*, 8(3), pp.239–260. Available from: https://www.tandfonline.com/doi/pdf/10.1080/13504620220145401.

Kosuth, M., Mason, S.A. and Wattenberg, E.V., 2018. Anthropogenic contamination of tap water, beer, and sea salt. PloS one, 13(4). Available: from:https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0194970

Kuntz-Duriseti, K., 2004. Evaluating the economic value of the precautionary principle: using cost benefit analysis to place a value on precaution. *Environmental Science & Policy*, 7(4), pp.291–301. Available from: https://www.sciencedirect.com/science/article/pii/S1462901104000619.

Laist, D.W., 1997. Impacts of marine debris: entanglement of marine life in marine debris including a comprehensive list of species with entanglement and ingestion records. *Marine debris*, Springer, pp.99–139. Available from: https://www.researchgate.net/profile/David_Laist/publication/235768493_Impacts_of_Marine_Debris_Entanglement_of_Marine_Life_in_Marine_Debris_Including_a_Comprehensive_List_of_Species_with_Entanglement_and_IngestiOn_Records/links/02bfe5136037d75c73000000.pdf.

Lancaster, K.J., 1966. A new approach to consumer theory. *Journal of political economy*, 74(2), pp.132–157. Available from: https://www.jstor.org/stable/pdf/1828835.pdf?casa_token=pommq7if0ccAAAAA:9W6rg6PZUNG6iN2e_sd9VLaBKszPZd7kUOD5LLmjAprkwCEn2XmhqPySN-SclOO9jR3dtdZVNmVv2Yqqe0YXaYA96KsBFExY4p4PA

Lancsar, E. and Louviere, J., 2008. Conducting discrete choice experiments to inform healthcare decision making. *Pharmacoeconomics*, 26(8), pp.661–677. Available from: https://link.springer.com/article/10.2165/00019053-200826080-00004.

Law, K. and Thompson, R., 2014. Microplastics in the seas. *Science*, 345(6193), pp.144– 145. Available from: http://science.sciencemag.org/content/sci/345/6193/144. full.pdf.

Lebreton, L., Egger, M. and Slat, B., 2019. A global mass budget for positively buoyant macroplastic debris in the ocean. *Scientific reports*, 9(1), pp.1–10. Available from: https://www.nature.com/articles/s41598-019-49413-5.

Lebreton, L., Slat, B., Ferrari, F., Sainte-Rose, B., Aitken, J., R.Marthouse, S.Hajbane, Cunsolo, S., A. Schwarz, Levivier, A. and Noble, K., 2018. Evidence that the great pacific garbage patch is rapidly accumulating plastic. *Scientific reports*, 8(1), p.4666. Available from: https://www.nature.com/articles/s41598-018-22939-w.

Lim, S.Y., Jin, S.J. and Yoo, S.H., 2017. The economic benefits of the dokdo seals restoration project in Korea: A contingent valuation study. *Sustainability*, 9(6), p.968. Available from: https://www.mdpi.com/2071-1050/9/6/968/pdf.

Logar, I., Brouwer, R., Maurer, M. and Ort, C., 2014. Cost-benefit analysis of the swiss national policy on reducing micropollutants in treated wastewater. *Environmental science & technology*, 48(21), pp.12500–12508. Available from: https://pubs.acs.org/doi/pdf/10.1021/es502338j.

Loomis, J.B., 2014. 2013 WAEA keynote address: Strategies for overcoming hypothetical bias in stated preference surveys. *Journal of Agricultural and Resource Economics*, pp.34–46. Available from: https://www.jstor.org/stable/pdf/44131313.pdf.

Lusher, A., Hollman, P. and Mendoza-Hill, J., 2017. *Microplastics in fisheries and agriculture*. Food and Agriculture Office of the United Nations. Available from: http://www.fao.org/3/a-i7677e.pdf.

Mensink, P. and Requate, T., 2005. The dixit–pindyck and the arrow– fisher–hanemann–henry option values are not equivalent: a note on fisher (2000). *Resource and Energy Economics*, 27(1), pp.83–88. Available from: https://www.sciencedirect.com/science/article/pii/S0928765504000314?casa_token=k-fRmxDMhp0AAAAA:B2gcRQNcBImROgWB_wpc2JeYuIWtuCk0U1OJvckFuCIVELQeGsNzle_XZiG5G4TbMDQuqE9i.

Morrissey Dr, K., 2019. Aligning ocean plastic pollution and human health a co-benefits approach. *Journal of Ocean and Coastal Economics*, 6(1), p.5. Available from: https://cbe.miis.edu/cgi/viewcontent.cgi?article=1090&context=joce.

Mouat, J., Lozano, R.L. and Bateson, H., 2010. *Economic impacts of marine litter*. Kommunenes Internasjonale Miljøorganisasjon. Available from: http://www.kimointernational.org/wp/wp-content/uploads/2017/ 09/KIMO_Economic-Impacts-of-Marine-Litter.pdf.

Murphy, J.J., Allen, P.G., Stevens, T.H. and Weatherhead, D., 2005. A meta-analysis of hypothetical bias in stated preference valuation. *Environmental and Resource Economics*, 30(3), pp.313–325. Available from: https://link.springer.com/content/ pdf/10.1007/s10640-004-3332-z.pdf.

Natural England, 2016. *Monitor of Engagement with the Natural Environment.* Available from: http://publications.naturalengland.org.uk/file/5257280994410496

OECD, 2006. *Cost-benefit analysis and the environment: Recent developments*. OECD. Available from: http://www.oecd.org/environment/tools-evaluation/36190261. pdf'.

Oosterhuis, F., Papyrakis, E. and Boteler, B., 2014. Economic instruments and marine litter control. *Ocean and coastal management*, 102, pp.47–54. Available from: https://www.sciencedirect.com/science/article/pii/S096456911400249X.

Orme, B., 2010. *Getting started with conjoint analysis: Strategies for product design and pricing research*. 2nd ed. Research Publishers LLC. Available from: https://www.sawtoothsoftware.com/download/techpap/samplesz.pdf.

Rakotonarivo, O.S., Schaafsma, M. and Hockley, N., 2016. A systematic review of the reliability and validity of discrete choice experiments in valuing non-market environmental goods. *Journal of environmental management*, 183, pp.98–109. Available from: https://www.sciencedirect.com/science/article/pii/S0301479716305709.

Robards, M., Piatt, J. and Wohl, K., 1995. Increasing frequency of plastic particles ingested by seabirds in the subarctic north pacific. *Marine Pollution Bulletin*, 30, pp.151–157. Available from: https://www.researchgate.net/profile/Martin_Robards/publication/222483960_Increasing_Frequency_of_Plastic_Particles_Ingested_by_Seabirds_in_the_Subarctic_North_Pacific/links/5a820ae80f7e9bda869f4fa1/Increasing-Frequency-of-Plastic-Particles-Ingested-by-Seabirds-in-the-Subarctic-Nort pdf.

Ryan, M., Gerard, K. and Amaya-Amaya, M., 2007. *Using discrete choice experiments to value health and health care*, vol. 11. Springer Science & Business Media. Available from: https://link.springer.com/content/pdf/10.1007%2F978-1-4020-5753-3.pdf.

Scasny, M. and Zvěřinová, I., 2014. *Stated-preference study to examine the economic value of benefits of avoiding selected adverse human health outcomes due to exposure to chemicals in the European union: Part ii: Fertility and developmental toxicity*. European Chemicals Agency. Available from: https://echa.europa.eu/documents/10162/13630/study_economic_benefits_avoiding_adverse_health_outcomes_2_en.pdf.

Shen, M., Huang, W., Chen, M., Song, B., Zeng, G. and Zhang, Y., 2020. (micro) plastic crisis: Un-ignorable contribution to global greenhouse gas emissions and climate change. *Journal of Cleaner Production*, p.120138. Available from: https://www.sciencedirect.com/science/article/pii/S0959652620301852.

Sudhakar, M., Trishul, A., Doble, M., Kumar, K.S., Jahan, S.S., Inbakandan, D., Viduthalai, R., Umadevi, V., Murthy, P.S. and Venkatesan, R., 2007. Biofouling and biodegradation of polyolefins in ocean waters. *Polymer Degradation and Stability*, 92(9), pp.1743–1752. Available from: https://www.sciencedirect.com/science/article/ pii/S0141391007001267.

Thompson, R.C., Moore, C.J., Vom Saal, F.S. and Swan, S.H., 2009. Plastics, the environment and human health: current consensus and future trends. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 364(1526), pp.2153–2166. Available from: https://royalsocietypublishing.org/doi/pdf/10.1098/rstb.2009.0053

Traeger, C.P., 2014. On option values in environmental and resource economics. *Resource and Energy Economics*, 37, pp.242–252. Available from: https://www.sciencedirect.com/science/article/pii/S0928765514000360.

Train, K., 2009. *Discrete choice methods with simulation*. Cambridge University Press. Available from: https://eml.berkeley.edu/books/choice2.html.

Train, K. and Weeks, M., 2005. Discrete choice models in preference space and willingness-to-pay space. *Applications of simulation methods in environmental and resource economics*, Springer, pp.1–16. Available from: https://www.repository.cam.ac.uk/bitstream/handle/1810/131583/cwpe0443.pdf?sequence=1

Vegter, A.C., Barletta, M., Beck, C., Borrero, J., Burton, H., Campbell, M.L., Costa, M.F., Eriksen, M., Eriksson, C., Estrades, A. et al., 2014. Global research priorities to mitigate plastic pollution impacts on marine wildlife. *Endangered Species Research*, 25(3), pp.225–247. Available from: https://www.int-res.com/articles/esr_oa/n025p225.pdf.

Vossler, C.A. and Watson, S.B., 2013. Understanding the consequences of consequentiality: Testing the validity of stated preferences in the field. *Journal of Economic Behavior & Organization*, 86, pp.137–147. Available from: https://www.sciencedirect.com/science/article/pii/S0167268112002661.

Watson, V. and Ryan, M., 2007. Exploring preference anomalies in double bounded contingent valuation. *Journal of Health Economics*, 26(3), pp.463–482. Available from: https://www.sciencedirect.com/science/article/pii/S0167629606001214?casa_token=mwt86eZF9vkAAAAA:Ni6w5FItIgWRMbGGT0Qots5Biq0QgnHi6bZ638d6wtfxkbw1xaprP7Va072AjAgSavlGRa0V#bib14.

Whitehead, J.C., Blomquist, G.C. et al., 2006. The use of contingent valuation in benefit-cost analysis. *Handbook on contingent valuation*, pp.92–115. Available from: http://www.appstate.edu/~whiteheadjc/ECO4660/PDF/Whitehead%20and%20Blomquist%20CH04.pdf.

Wit, W. de and Bigaud, N., 2019. No plastic in nature: Assessing plastic ingestion from nature to people. *World Wildlife Fund*. Available from: https://d2ouvy59p0dg6k.cloudfront.net/downloads/plastic_ingestion_web_spreads.pdf.

Zambrano-Monserrate, M.A. and Ruano, M.A., 2020. Estimating the damage cost of plastic waste in galapagos islands: A contingent valuation approach. *Marine Policy*, 117, p.103933. Available from: https://www.sciencedirect.com/science/article/pii/S0308597X19309820.